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Framework for assessing impacts of pile-driving noise from offshore wind farm construction on a harbour seal population

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ABSTRACT

Offshore wind farm developments may impact protected marine mammal populations, requiring appropriate assessment under the EU Habitats Directive. We describe a framework developed to assess population level impacts of disturbance from piling noise on a protected harbour seal population in the vicinity of proposed wind farm developments in NE Scotland. Spatial patterns of seal distribution and received noise levels are integrated with available data on the potential impacts of noise to predict how many individuals are displaced or experience auditory injury. Expert judgement is used to link these impacts to changes in vital rates and applied to population models that compare population changes under baseline and construction scenarios over a 25 year period. We use published data and hypothetical piling scenarios to illustrate how the assessment framework has been used to support environmental assessments, explore the sensitivity of the framework to key assumptions, and discuss its potential application to other populations of marine mammals.

1. Introduction

Growth in offshore wind generation is anticipated to play a major role in meeting the European carbon reduction targets that have been developed to mitigate potential impacts of climate change (Jay, 2011; Toke, 2011). In the North Sea, many proposed wind farm sites are on submerged offshore sandbanks, which also provide important habitats for marine mammals and seabirds. Previously, attention has focussed on the potential impacts of these developments upon bird populations (eg. Drewitt and Langston, 2006; Masden et al., 2010); however, several developments are in the vicinity of Special Areas of Conservation (SAC) that have been established to protect populations of marine mammals, such as harbour seals (*Phoca vitulina*) and bottlenose dolphins (*Tursiops truncatus*), under the EU Habitats Directive (92/43/EEC). Where developments have the potential to impact upon these species, Appropriate Assessments (AA) are required to establish that there will be no long-term impact on the integrity of these protected populations (Söderman, 2009).

There are three key potential impacts of offshore wind farm construction upon marine mammal populations. First, direct impacts of piling noise or other activities during the construction phase potentially causing direct injury or eliciting behavioural responses that could lead to displacement (Bailey et al., 2010; Brandt et al., 2011). Second, indirect impacts through long-term alteration of habitat that may, in turn, be either negative (through loss of habitat) or positive (though reef effects or changes in fishing activity) (Inger et al., 2009; Scheidat et al., 2011). Finally, disturbance or barrier effects resulting from operational turbines or maintenance vessels (Tougaard et al., 2009) may lead to displacement from areas or changes in movements. Given the high sound source levels resulting from pile-driving, the potential impacts that have been of greatest concern to stakeholders are the direct and indirect impacts of noise during construction (Dolman and Simmonds, 2010).

To obtain project consent, developers must provide information that allows regulators to conduct an AA to determine whether the proposal is likely to have a significant effect on the SAC's conservation objectives. Critically, this requires an assessment of whether the development may lead to long-term population change that would compromise the Favourable Conservation Status (FCS) of the wider population. However, whilst there is growing understanding that

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anthropogenic noise may affect individual behaviour in marine mammals (Brandt et al., 2011; Carstensen et al., 2006; Southall et al., 2007), the lack of guidance on how developers should assess the population consequences of disturbance from construction activities threatens to delay consenting decisions and efforts to meet 2020 carbon reduction targets.

To address this gap, we developed a framework for assessing population-level impacts of proposed wind farm construction on protected harbour seals using the Dornoch Firth and Morrich More SAC, within the Moray Firth, NE Scotland. The Moray Offshore Renewables Ltd (MORL) project comprises of three wind farms, with a combined output of 1.5 GW, located a minimum of 12 Nm from shore on the Smith Bank, leased under the Crown Estates Round 3 programme. The Beatrice Offshore Wind Limited (BOWL) project is a 1 GW development located adjacent to the MORL project within the 12 Nm limit, leased under the Crown Estates Scottish Territorial Waters (STW) programme. Construction of the projects is proposed to commence in 2015/16, which would allow both projects to be fully commissioned by 2020.

This paper provides an overview of the assessment framework that was developed to explore the long-term impact of different construction scenarios. Here, we illustrate the framework using pile-driving data collected in the Moray Firth during the installation of the two 5 MW Beatrice Demonstrator turbines (Bailey et al., 2010), and scenarios involving construction of a hypothetical wind farm at this site using analogous construction techniques. The development of this framework benefitted from a long history of research on the Moray Firth harbour seal population and information gained during the Beatrice Demonstrator Project. We therefore conclude by discussing how the framework can be developed to incorporate new data sources, and explore its potential use for other harbour seal populations.

2. General approach

Our general approach for assessing a development's impact on the SAC conservation objectives and the population's FCS involved three main elements (Fig. 1). First, available data and existing modelling frameworks were used to describe the spatial distribution of both harbour seals and pile-driving noise. Second, these data were integrated with available data on the potential impacts of noise to assess the numbers of individuals impacted. Finally, impacts on individuals were translated into changes in vital rates (fecundity and survival), and applied to a population model to predict longer-term population level impacts (considered here over a 25-year time scale).

Whilst these first two elements are routinely used in environmental assessments, uncertainty over the links between individual impacts and changes in vital rates has previously constrained efforts to model long-term population change. Frameworks for understanding population consequences of acoustic disturbance are being developed in response to the recommendations of a National Research Council Committee (NRC, 2003). While these approaches show great promise, empirical data to inform these links will not become available within the timescales required for consenting current rounds of offshore wind farms within the UK. Because AA of FCS must consider whether or not protected populations are maintaining themselves in the long-term, it was essential to develop an alternative transparent way of linking predicted individual impacts to vital rates. We therefore based these links upon expert judgement, while ensuring that the sensitivity of our population models to these assumptions could be fully explored. Similarly, the population modelling framework was required to permit exploration of potential interactions with other cumulative impacts (such as persecution or by-catch), and comparison of different development or mitigation scenarios. Crucially, while this initial framework was based upon the best available scientific data, it was designed to ensure that key parameters or relationships could be updated once new data become available.

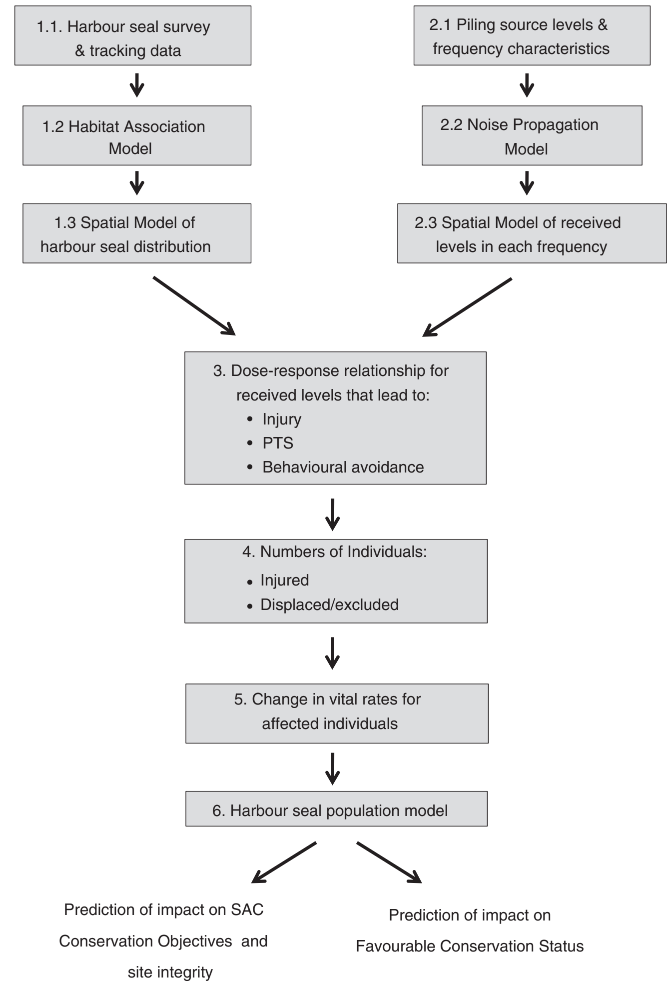


Fig. 1. Schematic of the approach proposed for assessing the impact of wind farm construction on the harbour seal Special Area of Conservation (SAC) and Favourable Conservation Status (FCS).

3. Components of the assessment framework

3.1. Characterising seal distribution

This element of the framework (see Fig. 1) requires information on spatial variation in the density of harbour seals across the region. Estimates of regional population size were available from annual counts at harbour seal breeding sites (SCOS, 2011), which were inflated to estimate the total number of individuals within the population (Thompson et al., 1997). In 2010, the mean haul-out count for the inner Moray Firth was 721, which represented a total population size of 1183 (95% CI = 1027–1329).

Information on the foraging distribution of seals was based upon tracking data from 37 individual seals, collected during a series of studies that were carried out in the area between 1989 and 2009 (Cordes et al., 2011; Sharples et al., 2012; Thompson et al., 1998). These data were used to model seal occurrence and habitat preference using a generalised additive model (GAM) to specify presence-absence across a 4×4 km grid, which revealed a significant relationship between seal presence and water depth, slope and distance to the nearest haul-out site. The results of this GAM were then used to predict the probability of seal occurrence in each of the 4×4 km cells across the Moray Firth (see Bailey et al., submitted for publication for further details). The percentage of the regional population in each cell within the Moray Firth was estimated by dispersing the whole population to produce a density surface in relation to the predicted importance of each cell (Fig. 2).

3.2. Characterising noise distribution

The predicted propagation of noise resulting from the piling operations required to install the wind turbine foundations was modelled using INSPIRE; a noise propagation model developed by Subacoustech Ltd. that has been widely used in environmental assessments for both renewable energy and oil and gas developments. This model uses a combined geometric and energy flow/hysteresis loss model suitable for pulsed noise such as impact piling to predict propagation in this shallow coastal environment. Comparison of INSPIRE model predictions with measured recordings from the Beatrice Demonstrator (Bailey et al., 2010) indicated that the model predictions for unweighted peak levels provide a relatively good fit of the measured data across the wider Moray Firth (Fig. 3).

Received sound levels were frequency weighted to account for the characteristics of harbour seal hearing. As detailed in Sec. 3.3, two different weightings were required. First, weighted peak sound pressure levels (dB_{ht} (species)) over single piling pulses (also termed "sensation level" (Yost, 2000)) were calculated based upon published data on the harbour seal audiogram (see Nedwell et al., 2007). Second, M-weighted sound exposure levels (SELs) were calculated based upon the approach proposed for all pinnipeds in water by (see Appendix A in Southall et al. (2007)). Spatial variation in received sound levels was expressed as a series of contours representing the point within which a particular threshold (e.g. 90 dB_{ht} (species) over a single pulse or an M-weighted SEL of 186 dB re 1 $\mu Pa^2\ s^{-1}$ over 24 h) was exceeded. Following discussion with the regulators and other stakeholders, estimates of M-weighted SEL contours were based on the assumption that animals would flee (at an average of 1.5 $m\ s^{-1}$) from the sound source rather than remaining stationary.

Outputs were generated as GIS shape files and used within ArcGIS to assess the maximum received sound pressure levels in each of the 4×4 km grid cells for which there were predictions of seal density (Fig. 2). An example showing the resulting dB_{ht} (harbour seal) values from INSPIRE for each grid cell is shown in Fig. 4.

3.3. Assessing impacts upon individual seals

The potential impacts of noise on marine mammals fall into three major categories; non-auditory injury, auditory injury, and behavioural (Richardson et al., 1995; Southall et al., 2007). These can each be further sub-divided depending upon the severity of the effect, as summarised in Table 1.

Traumatic non-auditory injury from loud sound sources at very close range is relatively well understood, and guidelines have been developed to mitigate against these risks (Dolman et al., 2009; Southall et al., 2007). While there is general agreement on this effect of sound, there is much more uncertainty over the mechanisms and received levels at which auditory injury and behavioural responses may occur. Drawing on the findings of a series of inter-disciplinary expert review groups, Southall et al. (2007) used available data to identify precautionary noise exposure criteria, weighting frequencies accordingly for different functional groups of marine mammals (M-weightings). Developed initially to support implementation of the US Marine Mammal Protection Act, many stakeholders now see Southall et al.'s (2007) interim criteria as the benchmark for environmental assessments in other parts of the world. However, whilst this can help identify noise levels that might cause injury, data on behavioural responses are so limited that Southall et al. (2007) only developed exposure criteria for behavioural responses to single pulses of sound. Given that a key concern during wind farm construction is behavioural disturbance from extended periods of pile-driving, regulators within the EU clearly cannot base their AA solely upon Southall et al.'s (2007) criteria.

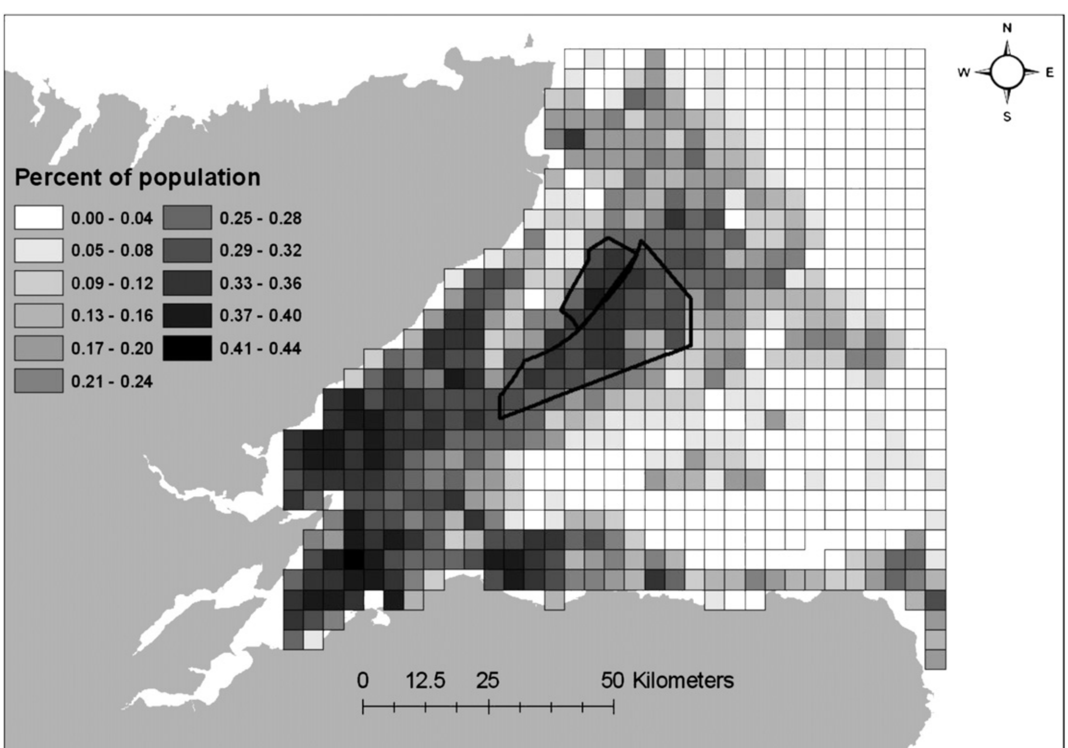


Fig. 2. At-sea distribution of harbour seals in the Moray Firth. Data are predictions from habitat association modelling based upon telemetry data from 37 harbour seals, and show the percentage of the population that is expected to be found in each of the different 4×4 km grid cells. The boundaries of the proposed MORL and BOWL wind farms are shown as a solid black line.

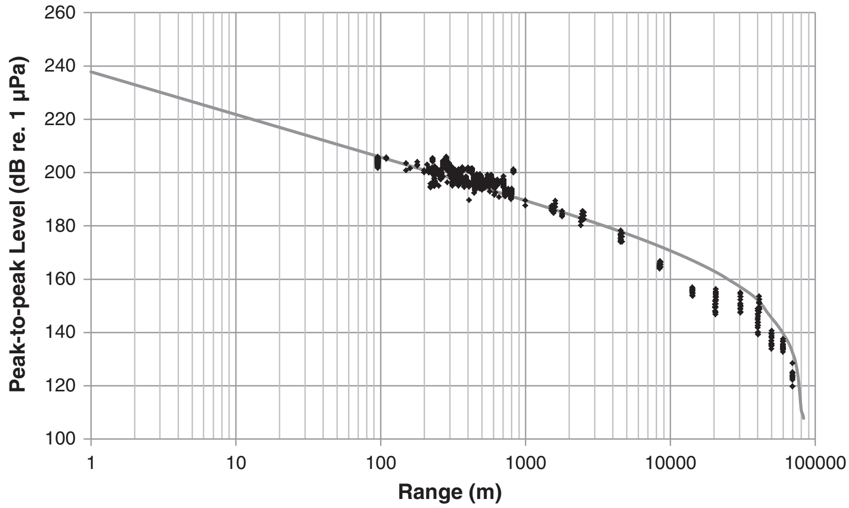


Fig. 3. The level of sound in unweighted peak to peak levels as a function of range in meters for a 1.8 m diameter pile at the Beatrice demonstrator site. Predictions from the INSPIRE model (solid line) are presented alongside measured levels (circles) from Bailey et al. (2010).

An alternative approach for assessing the impacts of anthropogenic noise, which focuses on behavioural responses, is the use of dB_{ht} (species) values as described by Nedwell et al. (2007). This approach builds upon standard procedures for assessing impacts of industrial noise upon humans, and uses information on each species' hearing ability to provide species-specific frequency weightings. This allows an assessment of the "perceived loudness" of the sound to the animal. Similarly, cognitive studies of marine mammals have estimated "sensation levels" that represent received levels, frequency-weighted according to the study species' hearing ability (e.g. Götz and Janik, 2010; Yost, 2000). However, these behavioural response criteria remain untested for marine mammals, and it is also recognised that behavioural responses are likely to be context specific (Ellison et al., 2012).

Given that there are uncertainties surrounding both Southall et al.'s (2007) M-weighted criteria and Nedwell et al.'s. (2007) dB_{ht} (species) criteria, our approach has been to estimate received levels using both metrics, and select the most appropriate metrics to assess different types of impact at the individual animal level. As described in more detail below, we use dB_{ht} (species) criteria to predict how many individuals will be displaced due to a behavioural reaction, and M-weighted criteria to predict how many individuals will be exposed to permanent threshold shift (PTS).

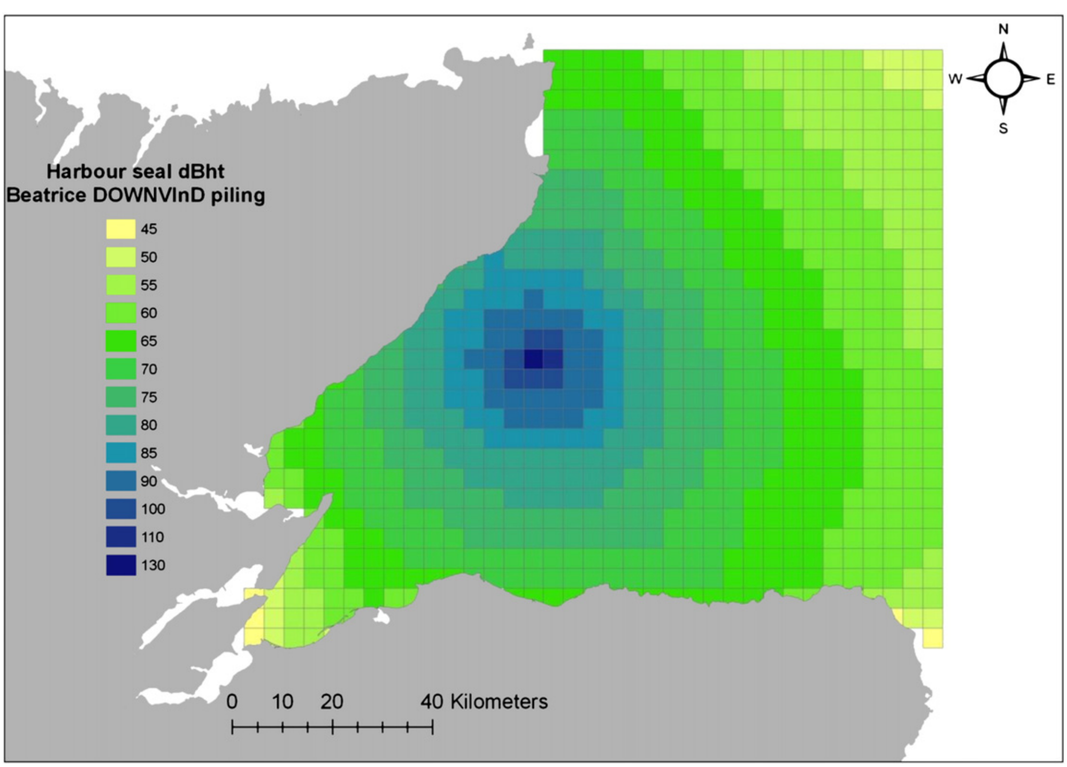


Fig. 4. Predicted received levels of pile-driving noise generated from Subacoustech's INSPIRE model, showing variation in received levels (dB_{ht} (harbour seal)) interpolated for each of the 4 × 4 km grid cells.

Table 1
Potential effects of noise upon marine mammals in order of severity.

Lethality and physical injury	
• Immediate death	Typically associated with rapid compression of air containing structures
• Physical Injury	
Auditory damage	
• Permanent Auditory Trauma/ (Permanent Threshold Shift)	Permanent elevation of hearing threshold
• Temporary Threshold Shift	Temporary elevation of hearing threshold.
Behavioural effects	
• Avoidance	See Table 4 in Southall et al. (2007) more detailed breakdown of behavioural effects
• Changes in foraging or social behaviour	

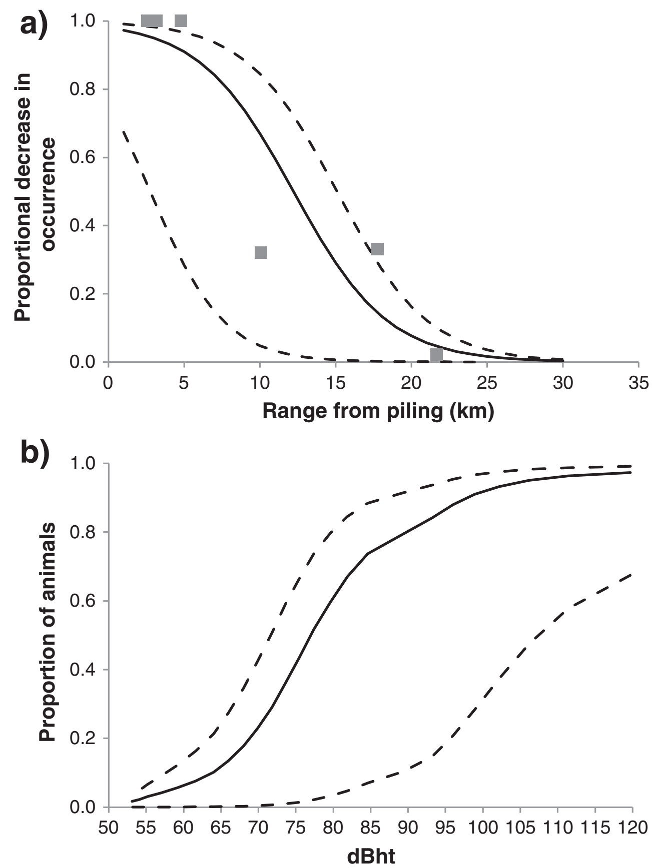


Fig. 5. a) Predicted relationship between range from the Horns Rev II piling operation and the proportional decrease in harbour porpoise occurrence (mean porpoise positive minutes from CPODs (from Brandt et al., 2011)) in the hour after the event; relationship for the line of best fit (deviance = 4.19; d.f. = 1; $P < 0.05$; Intercept = 3.9 (se = 2.77; Range = −0.32 (SE = 0.23)). The best fitted relationship is shown as a solid line. Standard errors were used to provide confidence limits around this relationship. However, because small sample sizes resulted in the upper bound showing almost no variation across the range of distances studied, we instead produced an upper bound for the relationship by weighting the line to include all data points. The lower bound is based upon the standard error of the coefficients. b) The relationship between dB_{ht} (harbour porpoise) and the predicted proportion of animals excluded from the area (using the upper, best and lower fitted relationship from Fig. 5a).

3.3.1. Risk of behavioural displacement

Behavioural responses may occur at many levels (see Table 4 in Southall et al., 2007). Here, we focus on behavioural responses that are likely to result in displacement from impacted areas, as we assume that lower levels of response will have only very weak links with vital rates. While this may underestimate impacts from more subtle behavioural changes, this should be balanced by our conservative assumptions about both the time that it takes animals to return to impacted areas and the consequences of behavioural avoidance (see Sec. 3.4).

Quantifying the levels of displacement is constrained by the absence of data on behavioural responses of harbour seals to known levels of multiple pulsed noise such as piling. However, Brandt et al. (2011) do provide data on changes in the occurrence of harbour porpoises at different distances from a piling event at Horns Rev II in Danish waters. These data were collected using moored echolocation detectors (C-PODs), and represent the difference between a baseline period and data collected during the hour after piling. We used these data from harbour porpoises as a proxy for harbour seals, and modelled the extent of the proportional change with distance from source by fitting a binomial relationship to the data (Fig. 5a). We then took published data on the size of the pile, together with information on local bathymetry, and used INSPIRE to estimate received sound pressure levels (using dB_{ht} (harbour porpoise) as a metric) at each of the C-POD sampling sites at Horns Rev II. The relationships from Fig. 5a were then used to predict the proportion of animals exhibiting a response at different received sound pressure levels using dB_{ht} (harbour porpoise) (Fig. 5b). In the absence of similar empirical data for harbour seals, we assume that this relationship holds for similar values of dB_{ht} (harbour seal). In other words, we assume that both species respond in a similar way to sound pressure levels within the range of frequencies that they are able to hear. We therefore use the relationship in Fig. 5b to predict the level of displacement of seals in each 4 × 4 km grid cell in relation to predicted sound pressure levels (using dB_{ht} (harbour seal) in that square.

3.3.2. Risk of auditory damage

Southall et al. (2007) provide interim noise exposure criteria for levels at which PTS becomes increasingly likely for the different functional groups of marine mammals. Given that one cannot experimentally induce PTS for ethical reasons, these noise exposure criteria for PTS-onset are conservative, and based upon assumed relationships between the relative levels of noise likely to cause Temporary Threshold Shifts (TTS) and PTS which, in turn, involves the use of proxy data from humans and other mammals.

We followed Southall et al.'s (2007) recommendation, and used an M-weighted PTS-onset threshold of 186 dB re 1 $\mu Pa^2\ s^{-1}$ for harbour seals. The increase in likelihood of PTS at higher levels of noise was then estimated by scaling up Finneran et al.'s (2005) dose response curve for changes in levels of TTS at different Sound Exposure Levels (SEL), where the probability of seals experiencing PTS increases from an SEL of 186 up to 240 dB re 1 $\mu Pa^2\ s^{-1}$; the point at which all animals are predicted to have PTS (see Supplementary material Fig. S1).

An alternative estimate of the number of individuals experiencing PTS was also generated using the SAFESIMM model, developed at the University of St Andrews as part of the Environmental Risk Management Capability (ERMC) (Donovan et al., 2012; Mollett et al., 2009). Originally developed to support the planning of Naval exercises, SAFESIMM is being adapted to support the management of marine renewable energy developments. SAFESIMM also used sound field data from INSPIRE, and a dose-response curve for PTS (also scaled from the TTS dose-response curve in Finneran et al. (2005)). However, it incorporates a more complex individual-based animal movement model when estimating accumulated sound exposure levels for different individuals in the population.

3.3.3. Estimating the number of individuals injured or displaced

To estimate the number of individual seals that would be exposed to injury, PTS or behavioural displacement, we used the relationships detailed above to assess the extent to which received noise levels in each 4 × 4 km grid square (e.g. Fig. 4) might impact the seals present in that grid square.

This process is illustrated in Fig. 6, where we estimate the number of harbour seals that may be displaced or suffer from PTS as a result of driving the Beatrice Demonstrator's 1.8 m piles. Fig. 6a presents the maximum received levels in each cell both using dB_{ht} (harbour seal) and fleeing animal M-weighted SEL as metrics. In this case, the 2010 estimated population of 1183 seals was distributed across grid cells in relation to the values shown in Fig. 2. We then predict the proportion of seals in each cell that would be displaced by the received levels in that cell as estimated using the relationships for behavioural disturbance and PTS, and sum these proportions to provide the total number of individuals affected.

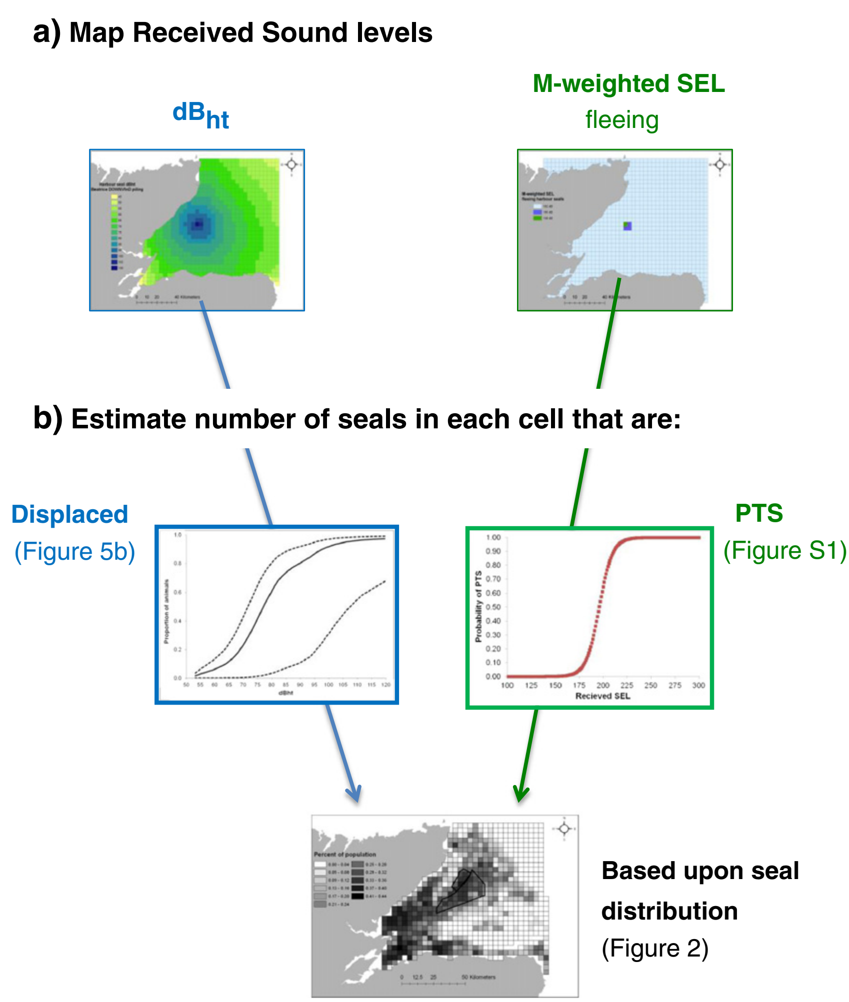
When modelling long-term population change, we also compared estimates of the number of animals experiencing PTS using this approach and using SAFESIMM.

3.4. Linking individual impacts to changes in vital rates

To model long-term population level effects of wind farm construction, we had to make a series of assumptions about potential changes in the reproductive and survival rates of those individuals that are predicted to be displaced or experience PTS. In the absence of empirical data, these assumptions were based upon expert opinion, where possible drawing from general ecological understanding or proxy studies. These assumptions were developed through a series of informal discussions and workshops with research scientists and other stakeholders. The rationale behind these assumptions is discussed below, and our proposed realistic worst-case scenarios are summarised in Table 2.

3.4.1. Death and non-auditory injury

Extremely loud sounds may have a direct effect on mortality at very close range. We assume that this will occur only at levels exceeding



c) Sum to estimate total number of animals affected

Behavioural Displacement			PTS
Upper	Best fit	Lower	Fleeing Model
690	511	55	56

Fig. 6. Schematic illustrating the approach used to assess the number of harbour seals from an estimated population of 1183 individuals that are displaced and vulnerable to Permanent Threshold Shift from an event involving the installation of two piles in 24 h.

Table 2
Assumed worst-case fitness consequences for individual seals exposed to different levels of pile-driving noise.

Effect	Consequence	
	Intermittent exposure	Constant exposure
Immediate death	Immediate mortality	Immediate mortality
Physical injury	Immediate mortality	Immediate mortality
Permanent threshold shift	25% risk of mortality	25% risk of mortality
Behavioural avoidance	Proportional reduction in reproductive success/and or juvenile survival	100% reproductive failure

220 dB_{ht} (harbour seal). INSPIRE modelling indicated that received levels from the installation of the 1.8 m piles only reached this level within <50 m. These are potentially major impacts at close range, but will likely be avoided by developing mitigation procedures routinely used during oil and gas surveys (JNCC, 2010). This assessment therefore focuses on the less direct effects of PTS and behavioural avoidance.

3.4.2. Consequences of PTS

Changes in hearing sensitivity might impact vital rates through changes in an individual's ability to forage, avoid predators or find mates. Harbour seals have extremely sensitive vibrissae which allow them to follow hydrodynamic trails from prey (Dehnhardt et al., 2001) and discriminate between different sized or shaped objects (Wieskotten et al., 2011). Given these capabilities, changes in hearing sensitivity from PTS appear less likely to have a direct impact on foraging ability compared with cetaceans. Where killer whale predation is high, a decrease in hearing sensitivity could increase the seals' risk of predation (Deecke et al., 2002). However, killer whales are rarely encountered in the North Sea and it seems unlikely that PTS would increase the risk of predation in this area. Finally, males make broad band vocalizations during their reproductive displays (Van Parijs et al., 1997), and these sounds may form cues when females are selecting males (Hayes et al., 2004). However, a reduction in hearing ability within part of the hearing range would seem unlikely to significantly reduce reproductive success, given that displays involve other visual and geographical cues and often occur in areas with relatively high levels of masking noise (Van Parijs et al., 1997, 1999). Nevertheless, there may be unknown fitness costs resulting from a decline in hearing ability that could affect reproduction or survival, and there was general stakeholder agreement that assessments of population level impacts should take account of this.

We addressed this by assuming that individuals experiencing PTS should be subjected to an additional mortality risk factor. In the absence of any data that could provide direct information on the mortality risk from PTS, we assumed that it was of a similar magnitude to the impact of old age. Information on age-specific survival in wild mammals is generally lacking but, typically, survival rates in the oldest age classes are 65–85% of adults in their prime (e.g. Beauplet et al., 2006; Loison et al., 1999). In our assessments, we assume that these costs are borne in the year after exposure, and impose an additional 25% risk of mortality on all animals that are estimated to have PTS. We assess the sensitivity of our results to this assumption by varying the severity of this additional mortality from 10 to 50%.

3.4.3. Consequences of behavioural displacement

We assume that the main behavioural impacts of noise are likely to result from avoidance of preferred foraging areas. The widespread distribution of harbour seals around the UK and other North Atlantic waters demonstrates that suitable foraging habitat is widespread, and their broad diet highlights that this is an extremely adaptable species. However, individual harbour seals also demonstrate high levels of site-fidelity (Cordes, 2011) and foraging ranges may be constrained around these favoured breeding and haul-out sites. Displacement could therefore lead to increased competition for food, greater energetic cost of foraging, or reduced foraging opportunities. As capital breeders, harbour seals build up energy resources throughout the year, feeding little or not at all during the breeding season. Given this life-history pattern, individuals should be relatively well buffered against short-term variability in prey availability. We therefore assume that the most likely impact of any reduction in an individual seal's overall energy balance will be a decline in reproductive success, which may manifest itself either by a reduction in the number of pups born or post-weaning survival of pups. Here, we make the conservative assumption that female harbour seals that are excluded from their foraging habitat will exhibit 100% breeding failure.

3.5. Modelling population level impacts

Population models have commonly been used to predict the future viability of age-structured vertebrate populations, including many species of pinnipeds. Such models are particularly useful for providing insights into the relative importance of specific management options or anthropogenic impacts but, as in our case, they rarely consider wider ecosystem consequences. In the context of offshore wind farms, population models have generally been considered in relation to assessments of the impact of bird strikes (Maclean et al., 2007).

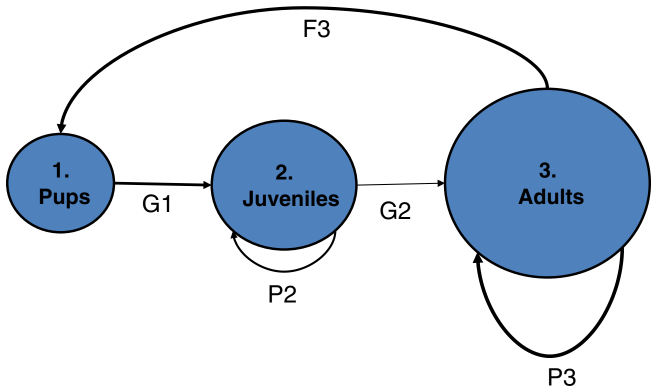
Recently, simple models have calculated the Potential Biological Removal (PBR) to provide managers with estimates of acceptable mortality from harvesting, culling or by-catch (Butler et al., 2008; Wade, 1998). This approach is suitable for supporting the management of activities that directly cause mortality, but is not adequate for assessing non-lethal anthropogenic impacts. Therefore, we adapted the stage-based matrix model previously used to estimate the impact of shooting on the Moray Firth harbour seal population (Thompson et al., 2007). By taking this approach, we were also able to explore potential changes in reproductive output or mortality that affect just certain age-classes or sexes. Furthermore, this approach allows us to incorporate cumulative impacts if, for example, licences are being granted to shoot seals within this management region (Butler et al., 2008)

We consider three life-history stages (Fig. 7) and model just the female component of the population, using an assumed equal sex ratio to inflate to total population size. Our baseline model uses the same input parameters as Thompson et al. (2007), supplemented by more recent analyses of photographic sightings of >150 individually recognisable harbour seals in Loch Fleet (Cordes, 2011), which is the harbour seal breeding site closest to the proposed development areas (Cordes et al., 2011). The input parameters used in the baseline model are presented in Table 3.

4. Application of the framework to assess construction scenario and sensitivity to key assumptions

We illustrate the use of this framework, and explore its sensitivity to key parameters, using a hypothetical construction scenario in which piling occurs simultaneously at two sites within the MORL development area over a four year period, starting in 2015.

The potential long-term impact of this hypothetical construction scenario on the Moray Firth harbour seal population was modelled by adjusting reproductive and mortality rates for the proportion of the population that were predicted to be affected by piling noise, as outlined in Table 2. We assume that any risk of immediate direct mortality can be avoided by mitigation, and that behavioural displacement occurs throughout the piling period (i.e. 100% of the year). The consequences of behavioural displacement were modelled as reduced reproductive success of displaced females by removing an appropriate number of pups (stage 1 seals in our model) in each of the construction years. To model the effects of PTS, we calculated the number of individuals that may suffer PTS, and removed 25% of these individuals from the population in each year. The resulting changes in population size over a 25-year



Stage	1	2	3	Stage	1	2	3
1	P1	F2	F3	1	0	0	0.88
2	G1	P2	0	2	0.7	0.66	0
3	0	G2	P3	3	0	0.19	0.97

Fig. 7. A life-cycle graph for the stage-classified single sex harbour seal model. Values for reproduction and survival rates, which represent the transition between stages, are taken from Table 3.

period were then compared with a baseline model in which no construction took place.

Table 4 presents data on the number of animals that are estimated to be displaced or experience PTS as a result of noise exposure from our hypothetical construction scenario. For displacement, we present values estimated using the upper, best-fit and lower behavioural response curves in Fig. 5b. For PTS, we present estimates made using our assessment framework (see Fig. 6) and estimates that have been produced by applying the same spatial distribution of received noise levels and initial animal densities within SAFESIMM.

As seen in Table 4, our estimates of the number of animals experiencing PTS were lower than those derived from SAFESIMM. Fig. 8 then compares the long-term population consequences of these individual impacts for the three different levels of behavioural displacement, and two different values for the severity of mortality resulting from PTS. In this case, we used SAFESIMM's estimates of PTS to illustrate worst-case scenarios. Fig. 9 compares long-term trends resulting from the use of these two different estimates of the number of animals experiencing PTS, and also explores the consequences of varying our assumption about the carrying capacity of the population. These comparisons highlight that the population trends appear to be driven largely by the baseline dynamics of the population. Although worse-case scenarios of impacts during construction could potentially lead to a short-term reduction in numbers, the long-term dynamics are not especially sensitive to uncertainty, for example, over the level of mortality resulting from PTS.

Table 4
Variation in the estimated number of individuals displaced and experiencing Permanent Threshold Shift (PTS) when using different curves to estimate behavioural resposnes, and when using our assessment framework and SAFESIMM to assess levels of PTS. Simulations use a hypothetical scenario involving simultaneous piling at two sites in the Moray Firth, and a population size of 1183 harbor seals.

	Number of individuals	% of population
Behavioural displacement		
Upper curve	690	58.7
Best fit curve	511	43.2
Lower curve	55	4.7
Permanent threshold shift		
Seal assessment framework	56	4.75
SAFESIMM	175	14.8

5. Assessing the significance of impacts

The spatial overlap of received sound levels and seal distribution, in combination with estimates of the impacts of noise exposure, potentially predicts a large number of seals being either displaced or experiencing PTS (Table 4). However, the population modelling used within the framework indicates that this should not result in long-term changes to the viability of this population. The use of different behavioural response curves and methods for estimating the impact of PTS resulted in variations in detail, but the common pattern at the population level was of short term reductions in abundance during and immediately after the construction period, followed by recovery that resulted in no observable difference between impact and baseline scenarios after 25 years.

A key issue when conducting Appropriate Assessments is the reversibility of any potential impacts on SACs. We therefore developed criteria that allowed us to use the outputs of our assessment framework to summarise the significance of impacts of different construction scenarios, taking account of both the magnitude and duration of those impacts (Table 5). The magnitude scale was guided by the principle that a high magnitude change should be measureable within the relevant timescale, taking account of background variation and sampling variability. At short durations (here a number of months), the magnitude of impacts can be assessed using estimates of the number of individuals in the population affected. At medium durations (a number of years), magnitude can be assessed by comparing the maximum difference between predicted population sizes for construction and baseline scenarios (typically at the end of the construction period). Finally, the long-term significance of developments can be assessed by comparing construction and baseline population sizes after 25 years.

Table 3
Values used for the life-history parameters and ecological characteristics used as input parameters in our baseline model.

Parameter	Values used	Source
Starting population size	1183	Estimate based upon SMRU 2010 surveys.
Age at first reproduction ♂	5 ♂, 4 ♀	Härkönen and Heide Jorgensen (1990).
Reproductive rate	88%	Cordes (2011).
Sex ratio	0.5	Boulva and McLaren (1979).
Density dependent variation in reproduction	Yes	Using equation 3 in Taylor and DeMaster (1993) to vary reproductive rate between maximum literature value at low population size (0.95 (Boulva and McLaren, 1979)) and a value of 0.1 at K (based on observed change in other pinnipeds (Fowler, 1990)).
Carrying capacity	2000	Conservative estimate based upon a value that is ~20% higher than the maximum abundance estimate in the last 20 yrs.
Pup/Juvenile mortality	30%	Harding et al., 2005; Härkönen and Heide Jorgensen, 1990.
Adult mortality	11% ♂; 3% ♀	Cordes (2011).

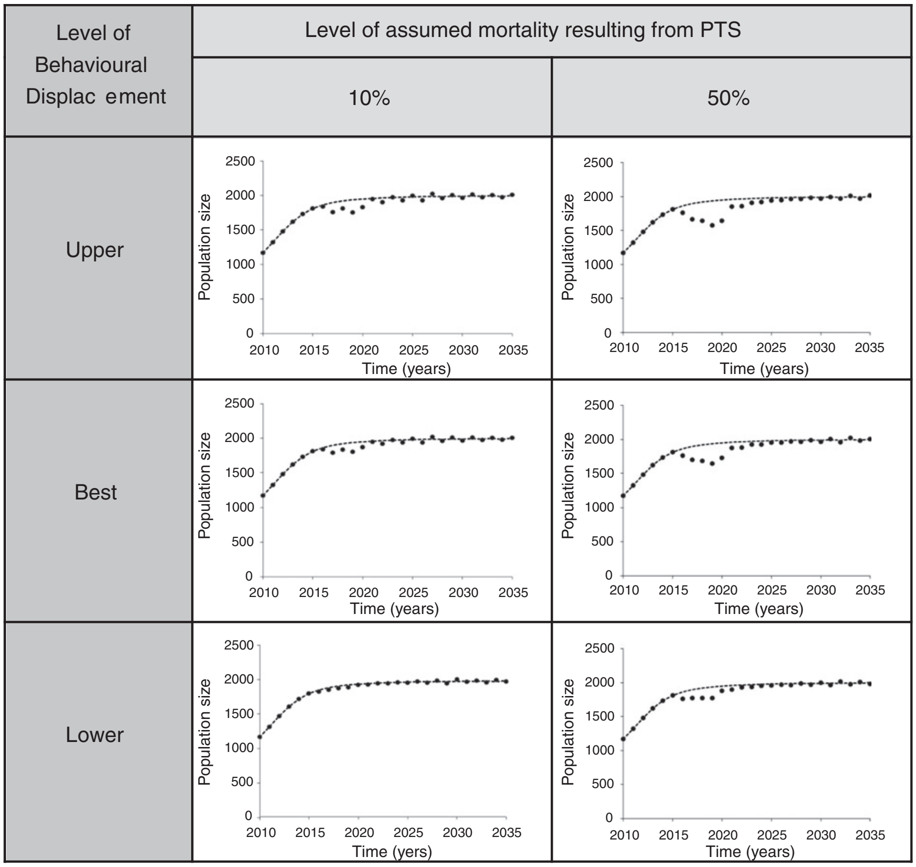


Fig. 8. Comparison of impact and baseline population trends using different behavioural response curves and severity levels for the mortality that may result from Permanent Threshold Shift (PTS). The percentages of the population affected in each of the 4 years of construction are taken from the hypothetical example in Table 4. Estimates of PTS are based upon numbers for the SAFESIMM modelling to provide a more precautionary approach.

6. Assessing and reducing uncertainty

Even for this relatively well-studied population, there remains enormous variation in the quality of data available to parameterise the different components of this framework and several key parameters have to be based upon expert opinion. Although some stakeholders would prefer to see additional data collected before decisions are made, this is impractical within the consenting timelines for the majority of proposed offshore wind farms. Instead, consenting decisions must be made utilising the information available so that regulators can achieve a balance between international agreements on climate change and nature conservation.

There are serious limitations in the amount of data available to assess the impacts of noise on marine mammal populations (Southall et al., 2007). Furthermore, even when data are available, they are often based upon small samples, with some key studies being based on single captive individuals. Consequently, the level of scientific uncertainty underpinning each element of our assessment framework for the Moray Firth harbour seal population varies considerably. A key aim in our approach was to ensure that uncertainty was explicitly recognised, and that the framework could be used to assess where to focus efforts on additional data collection. In response to stakeholder advice, we used the Intergovernmental Panel on Climate Change guidance upon the classification of uncertainty (Mastrandrea et al., 2010) to provide an indication of the relative confidence in different components of our framework. In the Supplementary information, Table S1 outlines the IPCC's recommended scale for characterising confidence in a dataset or assumption, based upon expert judgement. This scale is then used in Table S2 to summarise the confidence that we place in the different data available to us for use in this assessment framework, and Table S3 outlines the key assumptions that we made, together with an indication of the sensitivity of our results to each of these assumptions.

6.1. Seal distribution

The telemetry data available from Moray Firth harbour seals provided a relatively high quality dataset on foraging distribution, with consistent patterns seen over a twenty year period (Cordes et al., 2011). Nevertheless, sample sizes were still small when extrapolating to the whole population, and biased towards the summer period. This currently constrains our ability to compare potential seasonal differences in foraging area use. Additional telemetry tag deployments could address this and provide better estimates of contemporary distribution and winter use prior to assessments of any changes in distribution that may occur in response to construction.

6.2. Noise distribution

Despite differences in the approaches used for noise propagation modelling, underwater acoustics is relatively well understood and we have a high level of confidence in estimates of received noise levels. Here, we used the INSPIRE model, but the framework could be adapted to use alternative noise propagation models and other noise metrics. Measured data from the Beatrice demonstrator were similar to the predicted peak to peak levels that we modelled using INSPIRE. However, additional calibrated recordings made throughout the frequencies used by marine mammals are required to validate INSPIRE's far field predictions using dB_{ht} (species) as a noise metric. Such data would also be valuable for comparing the performance of different propagation models.

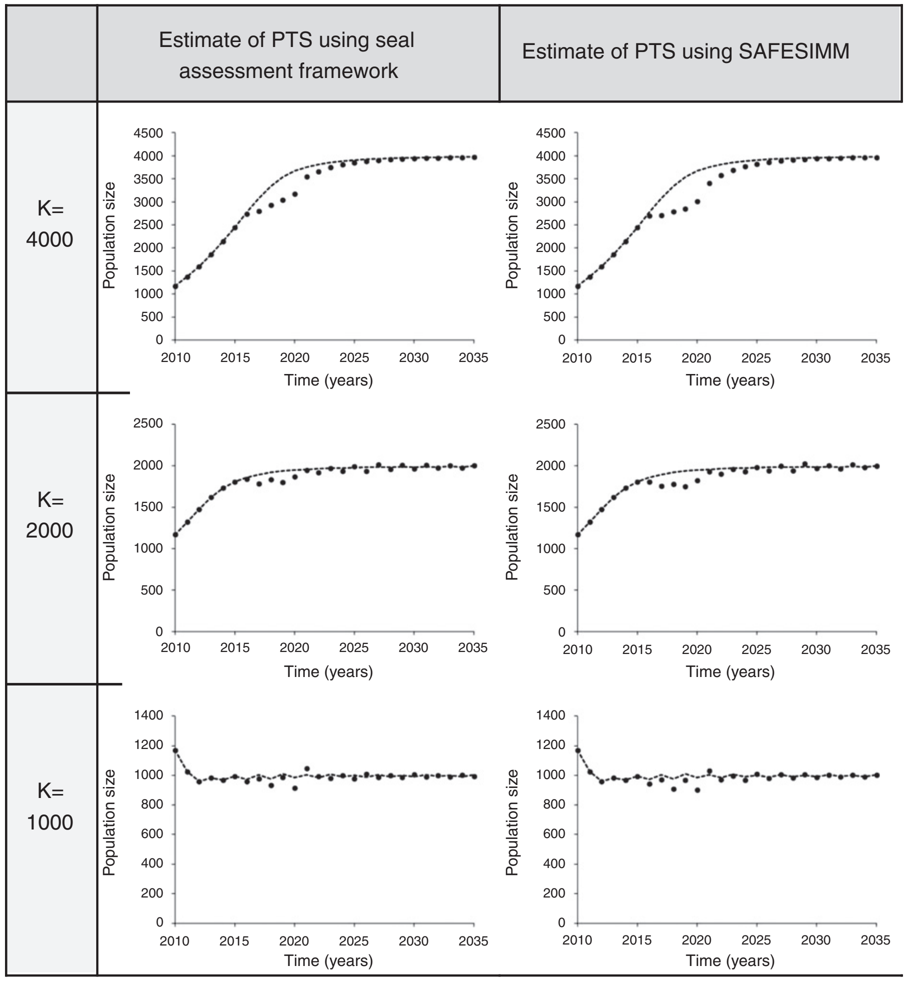


Fig. 9. Comparison of the effects of using different estimates for the number of individuals experiencing Permanent Threshold Shift (PTS) and of varying assumptions about the carrying capacity (K) of the environment. Mortality resulting from PTS is assumed to be 25% in all of these model runs.

6.3. Assessing impacts on individuals

There is far less certainty about the extent to which predicted noise levels may impact individual seals. The preliminary nature of the noise exposure criteria developed by Southall et al. (2007) highlights the evolving nature of understanding in this area. Planned research in the USA should provide additional data on TTS-onset to pulsed sounds such as pile-driving (Southall, Pers. Comm.) but this remains an area where it is difficult to obtain robust data with sufficient sample sizes. Studies of individual variability in the hearing thresholds in wild harbour seals should provide an additional tool for understanding issues. Recent studies of captive marine mammals have used measurements of auditory evoked potential (AEP) to assess hearing ability (e.g. Lucke et al., 2009) and this technique has excellent potential for use on wild animals; for example when individual seals are being caught and instrumented with tracking devices. In future, routine AEP tests during captures of wild seals should provide an important baseline to underpin future studies of changes in hearing ability over time.

Given the lack of data on marine mammal behavioural responses to different levels of pulsed noise, we used published data from Horns Rev II to provide an interim proxy for a dose-response curve. This is a first step, based on small sample sizes and a study of harbour porpoises rather than harbour seals. Furthermore, these data represent displacement for only a one-hour period after piling had ceased. There is a critical need for better data on recovery times after these displacements, particularly as these will affect the cumulative extent of displacement throughout a season of intermittent piling. Crucially, and as highlighted in Southall et al. (2007), there is an urgent requirement for studies which assess variation in levels of behavioural response in parallel with detailed characterisation of noise fields (Southall et al., 2012); ideally involving using a variety of different noise measurement metrics.

Our assessment framework applied the same dose–response curve for PTS that is used within SAFESIMM. Based on data from Finneran et al. (2005), this assumes that around 18% of animals exposed to Southall et al.'s (2007) PTS-onset criteria experience PTS, with the proportion gradually increasing at higher SELs. However, SAFESIMM

Table 5
Criteria used for predicting significance from magnitude of impact and duration.

Magnitude	Duration: Short (days)	Medium (construction years)	Long-term (25 yrs)
High (>20%) of population	Major significance (short term)	Major significance (medium term)	Major significance (long-term)
Medium (>10%)	Minor significance (short term)	Medium significance (medium term)	Medium significance (long-term)
Low (<10%)	Negligible significance	Minor significance (medium term)	Minor significance (long-term)

estimates of the number of seals experiencing PTS were higher. Further exploration is required to determine why these results differ, although it is likely to be due to differences in the way that the two approaches model how seals move away from loud noises. In the meantime, we take the more conservative approach and use estimates based upon SAFESIMM outputs in the AA.

6.4. Linking individual impacts to demographic parameters

Even with more certain data on the number of individuals displaced or experiencing PTS, there remains huge uncertainty over their subsequent consequences for individual fitness. It is these parameters that currently depend entirely upon expert judgement. Here, we use values that represent reasonable worst-case scenarios, but the modelling framework has been constructed so that these can be modified to explore sensitivity to variation in these values. This also allows us to explore where further research effort might best be placed. For example, there are clear limitations in carrying out further work to understand how variation in received noise affects the likelihood of PTS. Instead, it is likely to be most appropriate to use expert judgement to inform these parameters in the short term and, in the longer term, to directly assess relationships between noise exposure and key demographic parameters using the Population Consequences of Acoustic Disturbance (PCAD) framework developed by NRC (2003). As an interim measure, the approach could also be developed to more formally collate expert opinions (e.g. Martin et al., 2012) and sample from resulting parameter distributions within a stochastic modelling framework.

Here, the effects of PTS were realised through a change in mortality, but more complex individual-based models could be developed to incorporate more subtle effects of PTS, for example on female reproductive rates. The Moray Firth harbour seal population offers excellent opportunities to develop detailed PCAD studies to directly estimate whether such effects should be considered, and to test other assumptions made in this assessment process. Individually identifiable seals at the haul-out sites closest to proposed wind farms have been studied since 2006, providing estimates of survival and fecundity, while direct measures of pupping date and lactation duration provide information on year-to-year variation in female condition (Cordes and Thompson, 2013). Similarly, fine-scale tracking technologies could be used to gather data on how foraging behaviour is influenced by noise exposure, allowing changes in foraging success to be incorporated into future models. Combined with the realistic potential for field based measurements of hearing ability and noise exposure, the data collection that would support the development of these PCAD models could be integrated into construction monitoring. Ideally, these studies should also consider wider ecosystem consequences of wind farm construction, for example those resulting from effects upon habitats and prey populations (eg. see Inger et al., 2009).

6.5. Harbour seal population model

The final element of our framework involves a simple deterministic population model for this regional population of harbour seals. Initial analyses of the distribution of seals were conducted within ArcGIS, but the resulting grid based data could then be easily manipulated within a MS Excel framework. We used a stage-base population model within Excel using the Pop Tools add-in (http://www.poptools.org). This approach also allowed us to either include or exclude other factors such as the PBR-based quota of seals that may be removed by fishermen under licence by Marine Scotland (Butler et al., 2008). One advantage of this deterministic framework is its quick operation, which allows rapid exploration of different scenarios and model sensitivity, potentially in workshop situations with different stakeholder input. Like the PCAD models discussed above, future work would benefit from using stochastic models to incorporate uncertainty into model predictions.

7. Applicability of the framework to other marine mammal populations

This framework was developed to inform consenting decisions in NE Scotland, where potential impacts on local harbour seal SACs have been identified. However, consent of these and other Scottish Territorial Water and Round 3 sites will also depend upon AA for different populations of harbour seals.

The history of research on the Moray Firth harbour seal population has clearly been a great benefit in the development of this framework, but a lack of such detailed site specific data should not constrain the use of this approach for other regional harbour seal populations. While the temporal spread of telemetry data in the Moray Firth is unique, extensive tracking has been conducted in other parts of the UK over the last 10 years (e.g. Sharples et al., 2012), and these data are currently being used in broader-scale habitat models to characterise harbour seal foraging distribution around the UK. Similarly, although annual haul-out counts are made at only a few UK sites, a regular programme of surveys provides broad-scale data on abundance and trends in different UK regions (Lonergan et al., 2007; SCOS, 2011).

One concern is the extent to which less frequent surveys in other areas accurately reflect recent regional trends. This will be important to establish, as initial model runs highlight that predicted long-term trends are driven largely by the underlying baseline trend. When baseline conditions are favourable, harbour seal populations can grow rapidly as demonstrated by rapid recovery from major natural mortality events such as Phocine Distemper Virus outbreaks (Härkönen et al., 2006). In contrast, some Scottish populations have shown marked declines over the last decade (Lonergan et al., 2007) and added pressures from renewable developments may exacerbate these declines even where they are not driving them. A good regional time-series of annual haul-out counts is therefore an important pre-requisite if this framework is to be used in other areas. It is likely to prove more difficult to obtain comparable demographic data in other regions and, even where individual-based studies can be initiated, several years of intensive research will be required before robust survival estimates can be made. On the other hand, fecundity estimates could be based on other data sources, as for UK grey seals, which may be collected more easily at other sites over shorter periods. Alternatively, it is a common approach to "borrow" data from better studied populations, or even other species (e.g. Caswell et al., 1998), when developing population models. Such uncertainty should therefore not constrain the development of similar modelling frameworks for other harbour seal populations.

8. Conclusions

In an ideal world, assessments of the consequences of pile-driving or other industrial noise on protected marine mammal populations would be based upon a detailed understanding of dose-response relationships between received noise levels and changes in vital rates. This may become possible in the future through initiatives such as the PCAD programme, but alternative interim approaches are required to provide regulators with confidence that proposed developments will not significantly impact the long-term integrity of populations using SACs. We argue that this must involve some element of expert judgement, and that the framework we have developed for assessing impacts upon the Moray Firth harbour seal population illustrates how this can be achieved in a transparent and adaptable way. Whilst we recognise that this approach involves considerable uncertainty, this framework clearly documents which elements are based upon empirical data and outlines the rationale underlying parameters that are informed by expert knowledge. In the future, uncertainty could be quantified more formally within a stochastic modelling framework that samples from distributions representing the full breadth and weight of expert opinion. However, to provide assessments within the timescales currently demanded by regulators, we have instead dealt with this issue by selecting conservative estimates for individual parameters. Whilst this is appropriate for ensuring that the worst-case scenarios do not compromise long-term population viability, this conservatism accumulates through the framework. This leads to much more significant short term impacts than we anticipate are likely. It is therefore important that stakeholders recognise that this assessment framework is assessing worst-case impacts to meet the needs of the EU Habitats Directive, and that these require moderation where assessments of most likely impact are required under the EU Environmental Assessment regulations. Research and monitoring programmes should be carefully designed around key consented wind farms to explicitly test these assumptions, and inform the development of this framework to reduce uncertainty and conservatism in future assessments.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.eiar.2013.06.005.

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